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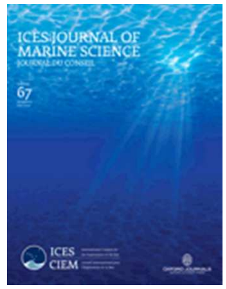

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Quantifying relative fishing impact on fish populations based on spatio-temporal overlap of fishing effort and stock density

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Abstract

Evaluations of the effects of management measures on fish populations are usually based on the analyses of population dynamics and estimates of fishing mortality from stock assessments. However, this approach may not be applicable in all cases, in particular for data limited stocks, which may suffer from uncertain catch information and consequently lack reliable estimates of fishing mortality. In this study we develop an approach to obtain proxies for changes in fishing mortality based on effort information and predicted stock distribution. Cod in the Kattegat is used as an example. We use GAM analyses to predict local cod densities and combine this with spatio-temporal data of fishing effort based on VMS (Vessel Monitoring System). To quantify local fishing impact on the stock, retention probability of the gears is taken into account. The results indicate a substantial decline in the impact of Danish demersal trawl fleet on cod in the Kattegat in

recent years, due to a combination of closed areas, introduction of selective gears and changes in overall effort.

Keywords: fishing impact, VMS, fish distribution, spatial modeling, Kattegat cod

Introduction

Marine environments and living resources are under an increasing focus of different policies aimed at achieving a healthy status of marine ecosystems, which include rebuilding depleted fish populations (WSSD, 2002; EC, 2008a, 2009). A critical element in the process of developing and implementing management strategies is to be able to evaluate their efficiency and monitor the progress towards achieving management objectives (Hall and Mainprize, 2004; Rice and Rochet, 2005). Fisheries management systems are often a compromise between multiple ecological and socio-economic objectives. Consequently, combinations of technical, spatial and other types of input (e.g., fishing effort) and output (e.g., total allowable catch) measures are implemented (Kjærsgaard and Frost, 2008; Prellezo and Gallastegui, 2008; Hornborg et al., 2012) and evaluating the effects of these measures on population dynamics is not trivial. Further, evaluating the effects of management measures based on stock dynamics is complicated by changes in growth, natural mortality, and year-class strength, which can counteract the effects of management measures or contribute to it (Pastoors et al., 2000; Eero et al., 2012a). Moreover, for depleted fish populations, catch information tends to become increasingly uncertain due to issues like over-quota discarding and other forms of poorly quantified catches (Cotter et al., 2004; Poos et al., 2010). Uncertain catch information often prevents reliable estimates of fishing mortality from stock assessments that are traditionally used to measure fishing impact on the stocks (Kraak et al., 2012). In such situations, alternative approaches such as those based on fishing effort and research survey information (e.g., Zhou and Griffiths, 2008) instead of fisheries removals from the stock are needed to quantify changes in fishing impact in response to management measures.

Spatio-temporal analyses of fisheries have greatly developed in recent years, supported by implementation of satellite-based vessel monitoring systems (VMS), which provide logbook-independent, high-resolution temporal and spatial information on fishing activities (Drouin, 2001; Lee et al., 2010). VMS data have frequently been used to describe the location of fishing grounds and the effects from spatial management measures (Dinmore et al., 2003; Murawski et al., 2005; Rijnsdorp et al., 2011). The measurements of fish distributions from research surveys have generally lower resolution both in time and space; nevertheless several modeling approaches that allow predicting local fish densities are being developed (Wood, 2008; Lewy and Kristensen, 2009; Maxwell et al., 2009). Combining the two types of analyses, i.e. fish distribution modeling and spatio-temporal analyses of fishing effort, could thus allow the overlap between the fisheries and the stock to be quantified. However, not all fishing techniques pose the same degree of risk to species or habitats (Witt and Godley, 2007). Thus, quantifying an overall impact from fisheries on an ecosystem component taking into account different elements of the pressure requires integrative analyses using multiple sources of information.

In this paper we develop a method for quantifying inter-annual changes in fishing impact on a fish population combining spatial modeling of species distributions based on trawl surveys with spatio-temporal analyses of fishing effort from VMS, and the results on retention probability at length from gear selectivity experiments available in the literature. The analyses are based on cod in the Kattegat, which is an example of a stock that has been severely depleted since the late 1990s (Svedäng and Bardon, 2003), and where uncertainties in the analytical assessment prevent evaluating recent changes in fishing mortality in response to management measures (Kraak et al., 2012). The approach presented in this study demonstrates an opportunity for fisheries management evaluations, gained from integrative analyses of information from research surveys and VMS.

Material and methods

Fisheries management measures for cod in the Kattegat

Due to a severely depleted state of cod in the Kattegat, several management measures have been applied in the Kattegat in recent years in order to reduce fishing mortality on cod. Total allowable catch (TAC) for cod has been reduced to 133 t in 2012, which is less than 1 % of the TAC 25 years ago. Besides TAC regulation, fishing in Kattegat is restricted by effort limitations. The amount of kW days for gear groups catching cod are subject to yearly reductions as long as the cod stock is below reference points defined in the management plan (EC, 2008b). However, the management plan offers possibilities for maintaining the allowable fishing effort if certain other measures are taken that reduce fishing mortality for cod, such as implementing closed areas or introducing selective gears (Kraak et al., 2012).

Large efforts have been devoted in recent years to improve both species and size selectivity of the trawls used in mixed fisheries in the Kattegat, to reduce the bycatch of cod and allow for continued exploitation of the economically most important species, Norway lobster (*Nephrops norvegicus*) and flatfish (Madsen and Valentinsson, 2010). In the Danish fisheries in the Kattegat, the usage of the exit window with square meshes at a minimum of 120 mm has been mandatory since 2008. Further, new trawls with sorting box (named SELTRA) with different designs and mesh sizes have been introduced (Madsen and Valentinsson, 2010; Madsen et al., 2010). Since 2011, the use of SELTRA trawls has become mandatory in the Danish fisheries in the Kattegat.

In 2009, as part of the attempts to rebuild the cod stock in the Kattegat, Sweden and Denmark introduced protected areas on historically important spawning grounds. The protected zone consists of four different areas in which the fisheries are either completely forbidden or limited to certain selective gears (Swedish sorting grid (Valentinsson and Ulmestrand, 2008) and Danish SELTRA with 300 mm mesh size in exit window) throughout part, or all, of the year (Figure 1).

Quantifying changes in fishing impact

The approach developed in this study for quantifying changes in fishing impact in response to management measures involves three steps:

i) modeling the distribution of fish based on survey data;

ii) mapping the distribution of fishing effort and estimating local fishing pressure based on VMS data and information on size selectivity of the gears used;

iii) estimating annual changes in fishing impact on the stock by overlaying the spatial and temporal distribution of fishing pressure and the stock.

The following sections describe each of the three steps in further detail.

Modeling cod distribution and related uncertainties

Analyses of the distribution of cod in the Kattegat were based on catch from research trawl surveys conducted in the 1st, 3rd and 4th quarter of a year. Time series from six surveys were available (Table 1), covering between 20 and 80 stations per year each (see Supplement A for distribution maps of survey stations). These surveys are also used in stock assessment of cod in the Kattegat in ICES where additional information including time series of catch per unit of effort from the surveys can be found (e.g., ICES, 2012).

The relative cod density was modeled using a Generalized Additive Model (GAM) of the catch in numbers (C) of cod in a size class (10–24 cm, 25–39 cm and ≥40 cm) by haul as a function of position, depth, year and survey:

$$C \sim \text{offset}(\log(\text{effort})) + \alpha + f1(\text{longitude, latitude}) + f2(\text{depth}) + f3(\text{year}) + survey + \varepsilon$$

where $f1$, $f2$ and $f3$ are smoothing functions and *survey* is a factor.

For the 4th quarter, where the data included only four years, the year effect was modeled as a factor. Haul duration was applied as effort variable for the IBTS, BITS and Danish sole survey (Table 1) where the same gear is used within each of these surveys. In the Danish/Swedish cod survey, different sizes of trawls are applied which was taken into account using swept area (between the doors) as a measure of effort for each haul.

The GAM analyses were conducted using the R package "mgcv" (www. r-project.org; Wood, 2006). Smoothing terms used penalized thin plate regression splines (Wood, 2003), where the effective degrees of freedom ('knots') associated with smoothing was selected as part of the model fitting (Wood, 2006, 2008). The upper limit of the number of knots had to be specified, but the choice of an upper limit is generally not critical (Wood, 2006). As a default, we have used the upper limits of knots suggested by the software. However, when the effective degrees of freedom for a model term were estimated close to the upper limit, further analyses were made to select the number of knots. This involved examining the distribution of deviance residuals in relation to explanatory variable, and changes in residual pattern depending on the number of knots applied.

For all analyses, non-significant model terms were removed from the final model. The negative binomial distribution and logarithmic link function were used to model catch numbers. The logarithm of effort was used as offset variable. Regression results of the GAM analyses are provided in Supplement A.

The uncertainty in the predicted relative density of cod was estimated from parametric bootstrapping. One thousand replicate parameter vectors from the fitted models, extracted such that their variance and co-variance were maintained, were used to predict the sum of densities (index) of cod for a 0.01^{o} longitude x 0.01^{o} latitude grid. From the bootstrap replicates the mean and variance of the total abundance index were calculated. The bootstrap replicates were also used to calculate the proportion of the stock within and outside the Areas 1–3 (Figure 1), which are partly or entirely closed for fisheries.

In the model it is assumed that the relative stock distribution (*f1*(longitude, latitude)) is independent of year. This assumption is based on the analyses of centre of gravity, performed on each survey time series, which showed a variable distribution from one year to the next, however no significant change over the year range included in modeling cod distribution (see Supplement A for details). No up to date survey information was available for quarter 2, so cod distribution in this quarter was assumed similar to that in quarter 1.

<u>Distribution of fishing effort by gear type</u>

Fishing effort of the Danish fleet in the Kattegat was analyzed for the period 2008–2011. Proxies for local fishing effort were based on vessel positions obtained from VMS, similar to the procedure applied in earlier studies (e.g., Deng et al. 2005; Lee et al., 2010; Rijnsdorp et al., 2011). The information on the level of total effort by fleet was derived from log-books, combined with the spatial distribution of effort represented by VMS data. VMS records with vessel speed of 2–4 knots were classified as fishing activity and combined with vessel log-book data by fishing trip. The trips were allocated to gear and mesh-size according to the information provided in the log-books, including the information on vessel engine power (kW). Coupling of VMS and log-book data was done following the methodology described by Bastardie et al. (2010), where the technical details can be found. VMS data are available at high temporal resolution (1 ping per hour), but were aggregated to quarterly values for the purpose of this study, to match the temporal resolution of cod distribution from survey analyses.

The vessels using the mesh size of 70–99 mm (the TR2 gear in EU regulation) is by far the most important part of demersal fisheries in the Kattegat (Table B1 in Supplement B). Therefore, effort from only this fleet (TR2) is included in the analyses of fishing impact. Until 2011, only gear group and mesh size were provided in the log-books, whereas specifying the exact rigging of the trawl impacting on cod avoidance was not mandatory. In the calculations, it was assumed that in the areas and periods where SELTRA 300 has been required by legislation, this gear has been used. Further, the exit window with square-meshes at 120 mm was considered as a default gear used since February 2008. Before that, the standard 90 mm gear was assumed to have been used. Vessels fishing (illegally) in the permanently closed area (Area 3 in Figure 1) were assumed to have used the default gear (120 mm exit window) or the gear type noted in the log-book if available.

VMS is only mandatory for vessels over 15 m in length, which is roughly about 60 % of the Danish effort in TR2 segment in the Kattegat. The effort distribution by gear of the fleet equipped with VMS was raised to the total effort of the TR2 fleet based on the proportion of total national effort by year and quarter coming from vessels with VMS. It is thereby assumed that large and small vessels have the same use of selective gears and the same spatial fishing pattern. The resulting distribution of the effort by year, area and gear type, used in further analyses of fishing impact on cod is presented in Table B2 in Supplement B.

Estimating fishing impact on cod

Fishing impact (I) was defined as:

$$I_{lon,lat,year,qrt,gear,size} = D_{lon,lat,qrt,size} \times E_{lon,lat,year,qrt,gear} \times R_{qrt,gear,size}$$

where,

D – relative stock density, i.e. the proportion of the stock (at size) in a given position (longitude, latitude) in a grid of 0.01°×0.01°, in a given quarter (qrt). Stock density was predicted from the fitted model of cod distribution.

E – fishing activity represented by the number of VMS records corresponding to fishing activity times the engine power (kW) raised to the total nominal effort of the fleet. Effort was calculated for each position (longitude, latitude) in the grid of 0.01°×0.01° by year, quarter and gear. The high number of VMS recordings, around 60000 per year, made it possible to use the observed effort within the specified grid directly as an unbiased estimator of effort.

R – Retention likelihood of cod at size group, derived from gear-specific selection (S) curves (Frandsen et al., 2009; Madsen and Valentinsson, 2010) and length distribution within each size group of the cod population derived from surveys.

$$R_{qrt,gear,size} = \sum_{l=first\ length_{size}}^{last\ length_{size}} \left(S_{gear,l} \times N_{qrt,l}\right) \Bigg/ \sum_{l=first\ length_{size}}^{last\ length_{size}} N_{qrt,l}$$

It was assumed that the number (N) of cod at length (l) caught during surveys represents the size distribution of the stock in the sea.

Local fishing impact was assumed to be proportional to local fishing pressure (combination of the local fishing effort and selectivity of the gear used) and cod density. The estimates of local fishing impact were subsequently aggregated to an overall estimate of fishing impact for cod in the Kattegat for each year from 2008 to 2011. The units for fishing impact are arbitrary, and the estimates were only used to quantify relative changes in fishing impact in the period from 2008–2011.

To quantify the effect of uncertainties in cod distribution on the relative change in fishing impact, bootstrap replicates of stock distribution were used as a basis for calculating one thousand sets of changes in fishing impact, from which the confidence limits were derived.

Results

Stock distribution

The results of modeling the distribution of cod in the Kattegat by size group and quarter show that in the 1st quarter, 10–24 cm cod (mainly age group 1) is relatively dispersed with the highest concentrations in the north-western Kattegat (Figure 2). In contrast, the density of larger cod is highest in the deeper part of the eastern Kattegat and north of the Sound, i.e. in the areas covered by fisheries closures (Areas 2 and 3 in Figure 1). In the 3rd quarter, the highest concentrations of cod in all size groups were found in the north-eastern Kattegat, including the partially closed area (Area 2), where selective gears are mandatory. In the 4th quarter, 10–24 cm cod were mainly found in the north-western Kattegat, whereas larger cod were distributed more southerly with the highest densities in the partially closed area (Figure 2). The standard deviations of the density estimates generally follow the densities; however, the areas with low densities and few observations, e.g. close to the coast line, have higher variations (Supplement A). The uncertainties of abundance index derived from bootstrap replicates show the lowest uncertainty for quarter 4 (e.g., Coefficient of Variation at 0.07 for the 25–39 cm group), slightly higher for quarter 1 (e.g., CV at 0.16 for the 25–39 cm group) and a high uncertainty for quarter 3 (e.g., 0.28 for the 25–39 cm group). For all quarters, uncertainties are highest for the larger, ≥ 40 cm, cod. Further details on uncertainties in cod distribution are presented in Supplement A.

The estimates of proportion of the stock within a given area (Table 2) show that about half of the adult cod (above 25 cm in length) population is found within the closed areas in quarter 1, which corresponds to high local cod densities (Figure 2). In other periods, the proportion of the stock within closed areas is lower, and is generally lower for small cod (10–24 cm) compared to larger individuals (above 25 cm). The uncertainties associated with proportions of the stock within given areas show similar patterns as the uncertainties in total stock distribution, i.e. uncertainties are rather low for quarter 4 and 1, but high for quarter 3; especially for the ≥ 40 cm size group (Table 2).

Catch position and water depth were significant terms in nearly all models, explaining significant proportions of the variation in survey catches in the Kattegat. Water depth was not significant in the models for quarter 3 (Table A1 in Supplement A).

Effort distribution

The main part of the Danish demersal fisheries in the Kattegat takes place on fishing grounds in deeper parts of the central and eastern Kattegat (Figure 3). The total effort of TR2 fleet does not show clear trends over 2008–2011; however the effort in 2011 is close to 20 percent lower compared to the effort in previous year (Table B2 in Supplement B). The spatial distribution of fishing effort shows pronounced changes in the years 2008–2011. The introduction of closed areas in the Kattegat in 2009 resulted in a westwards relocation of the effort in the 1st quarter (Figure 3), when all fisheries in Areas 2 and 3 (Figure 1) were banned due to cod spawning closure. In 2009, fishing effort decreased substantially also in the 2nd and 3rd quarter in the partially closed area (Area 2), where selective gears are required. An opposite pattern was observed in 2010, when most of the Danish fishery by TR2 fleet was concentrated in the partially closed area. However, this behavior of the fleet was only temporary, as almost no fishing activity was recorded in this area in 2011. In the 4th quarter, changes in fisheries distribution since 2008 have generally been less pronounced compared to the other quarters of a year. Some VMS activity classified as fishing was recorded in the permanently closed area in 2010, while in other years since 2009 the activity in this area has been insignificant.

Changes in fishing impact on cod

The fishing pressure overlaid with cod distribution shows that the overall impact from the Danish TR2 fleet has been reduced for all size groups of cod in the period from 2008 to 2011 (Table 3). The fishing impact in 2011 was estimated to be 45, 40 and 45 percent of the impact in 2008 for cod size groups 10–24, 25–39 and ≥40 cm, respectively, i.e. a reduction of around 60 percent. The strongest decline in fishing impact for cod larger than 25 cm occurred in 2009 (around 30 percent) followed by a modest reduction in 2010 and a higher reduction in 2011 (Table 3). The reduction in fishing impact on cod was largest in the areas subject to permanent or partial closures; however, a decline in fishing impact was estimated also in the areas outside of closures due a general change to more selective gears. In contrast, in the seasonally closed area (Area 1 in Figure 1), the fishing impact was estimated to have increased in 2009–2010 in relation to 2008 (Table 3).

Box and whisker plots of the distribution of bootstrap replicates of the relative fishing impact (Figure 4) show statistically significant changes in the impact for all size groups of cod in the period 2008–2011. For all combinations of year and size groups, the uncertainties in the annual relative fishing impact were estimated with a CV at around 5%, with a maximum at 6.6% (25–39 cm in 2011) and a minimum at 1.9 % (10–24 cm in 2010).

Discussion

Uncertainties related to stock distribution

An overarching assumption in the approach applied in this study is that local fishing impact is proportional to the sum of product of local fish density, local fishing effort and size selection of the gears applied. Each component in this combination is associated with uncertainties. In our study, we have focused on the uncertainties associated with cod distribution and incorporated these in the analyses of fishing impact. Cod is a mobile species with spawning and feeding migrations (Hüssy et al., 2009), and homing of spawners is a primary mechanism of stock separation (Rindorf and Lewy, 2006; Svedäng et al., 2007). Estimating local cod densities obviously requires good temporal and spatial coverage of surveys. This is clearly demonstrated in our analyses where good survey coverage in quarter 1 and 4 resulted in relatively low uncertainties in fitted stock distribution, whereas the uncertainties were much higher for quarter 3 due to fewer hauls in surveys. The low uncertainty in cod distribution in seasons with good survey coverage supports the assumption that no consistent changes in distribution have taken place in the analysed period, which is confirmed by the analyses of centre of gravity (see Supplement A). Therefore, the available information for the spawning season in winter and spring and for the feeding period in autumn likely captures the main patterns in cod distribution.

An important prerequisite for estimating local fishing impact is that the survey data used to estimate fish densities cover the same areas and habitats as the fisheries. This is the case for Kattegat as the TR2 fleet mainly targets Norway lobster on soft bottoms, which are also covered by trawl surveys. In some other areas, fishery may be concentrated on hard bottom types, not sampled by research surveys, as described for cod in the North Sea (Wieland et al., 2009).

GAM-based spatial modelling is recognized as a tool for producing distribution maps of fish and zooplankton, though not frequently applied on data from trawl surveys (Murase et al., 2009). An advantage of the GAM method is that it allows the predicted catch (density) to form a likely smooth surface of the spatial distribution. Hence, it is natural to use non-parametric regression technique to analyse survey data (Cadigan and Chen, 2001). Further, GAM analyses can incorporate interactions between animal distributions and environmental factors (Swartzman et al., 1999; Winter et al., 2007; Wood, 2008). The process of building distribution models is suggested to include cross-validation of the models across years (O'Brien and Rago, 1996). Also, inter-annual changes in marine fish distribution can take place, in relation to changes in environment or in stock size (Burrows et al., 2011; Overholtz et al., 2011). In our analyses we were not able to detect significant changes in cod distribution within the time series used in the analyses. However, if cod will recover in the future, this may lead to a shift in the distribution of the spawning stock. For example, some historical spawning sites may currently not be utilized by cod possibly due to eradication of local sub-populations (Svedäng et al. 2010), but which might be recolonized at higher stock sizes. To capture such changes will require continued good survey coverage and regular updates of cod distribution.

Uncertainties in estimating fishing effort based in VMS

The VMS recordings corresponding to fishing activity are usually identified based on vessel speed (e.g., Needle and Catarino, 2011). However, some bias may be introduced in this classification as a fishing vessel may slow down due to a number of other reasons than fishing, such as approaching or leaving port, setting gear, being in the proximity of other boats, or due to weather conditions (Mills et al., 2007). In our investigation, we have used the method suggested by Bastardie et al. 2010 to identify 'fishing activities' and to link VMS fishing recordings with log-book data. This method has been developed and tested for the Kattegat /Skagerrak area and it shows that some misclassification of both vessel activity and fleet segment might occur. Also, only the larger vessels (above 15 m in length in the Danish fisheries) are equipped with VMS, and scaling their fishing pattern to the entire TR2 fleet may introduce some bias in the estimates of local fishing effort. However, the main fishing grounds for trawlers in the Kattegat are expected to be similar for large and small vessels. This is related to the characteristics of the Kattegat area, which is generally shallow with an extensive shelf (~10m depth) covering most of the western part, with a deeper trench (>90m) running along the Swedish coast in the eastern part of the area.

Compared to the uncertainties in catch information (Cotter et al., 2004; Poos et al., 2010) traditionally used in measuring fishing impact, the effort information based on VMS can likely be considered as relatively more accurate. For example, even some illegal fishing activities, like fishing in closed areas are captured by VMS data and can be taken into account in the analyses of fishing impact. Thus, although the absolute level of effort can be biased due to the above-mentioned uncertainties, the relative changes in fishing effort between years that is our main focus in this study, are probably not seriously affected by these uncertainties.

Uncertainties in gear selectivity

Estimating selectivity of a fishing gear usually involves experiments at sea, recapturing the fish escaping through the meshes. However, some methodological uncertainties may be introduced in this process (Millar, 2010) and small changes in gear design can lead to very different conclusions about its selectivity. We have not attempted to include uncertainties in trawl selection to our analyses of fishing impact. This is partly because some of the uncertainties of the parameter estimates from the trawl selection experiments are not available in the literature, and partly because the selection achieved during commercial fishery might differ considerably from the theoretical estimates. In addition, uncertainties in selection apply both to the reference gears used in 2008 and to the new more selective gears, which further complicates taking these uncertainties into account. The selectivity parameters used in the analyses of fishing impact can be revised when new information becomes available.

Applicability of the approach investigating spatio-temporal changes in fishing impact

Reliable estimates of fishing mortality or exploitation rate (often expressed as F and U) are not available for cod in the Kattegat and we did not intend to estimate these conventional measures of fishing impact in our study. The approach we developed can produce proxies for changes in fishing mortality under an assumption that local fishing impact (mortality) is proportional to the product of local fish density, fishing effort, and gear selectivity. This may not be valid in all cases (Poos and Rijnsdorp, 2007; Quirijns et al., 2008; van Oostenbrugge et al., 2008). Further, the method assumes that fish are re-distributed in an area after a trawling activity has taken place, which makes the approach not suited for sedentary and low mobility stocks. Evaluations of fishing impact on sedentary species like clams or Norway lobsters will require other approaches (e.g., Bustamante et al., 2010) or at least sufficient data to regularly update stock distributions.

An advantage of using an effort-based measure of fishing impact instead of fishing mortality rate from stock assessment is that the former approach potentially allows separating the effects of individual management measures, affecting the components of fishing impact explicitly considered in the analyses. Nevertheless, separating the effects of individual management measures is challenging when different measures act in combination and enforce one another (Murawski et al., 2000; Madsen and Valentinsson, 2010). The Kattegat cod example illustrates that implementation of closed areas with exemptions for selective gears created incentives for using such gears, to gain access to the areas otherwise closed for fisheries. This explains the large inter-annual changes in location of fishing effort, moving out of the partially closed areas in the first year of closure (2009), however returning a year after (2010) with new selective gears. The reasons for subsequently low fishing activity in the areas requiring selective gears in 2011 are not known to us.

The explicit consideration of spatial scales in this approach to fishing impact is advantageous, given the spatial heterogeneity and dynamics of marine ecosystems

(Lorenzen et al., 2010; Eero et al., 2012b), which sets an increasing focus on spatial aspects in marine management. The Kattegat example demonstrates expected low fishing impact in closed areas after their implementation, however an increase in fishing impact in bordering areas (Area 1, Table 3) in some years. This is a commonly seen phenomenon and area closures are therefore generally considered as just one element in a broader package of fisheries management measures (Hilborn et al., 2004). It is also apparent from our analyses that closed areas in the Kattegat are mainly a tool to protect the spawning stock, as small cod is generally wider distributed in the entire area, and fishing impact on those can probably best be regulated by selective gears. For example, the reduction in fishing impact on larger cod in 2009 was mainly due to closed areas, whereas very little effect was found on small cod, which are to a large extent distributed outside the closures. The approach developed in this study could potentially also be used to explore the effects of different locations for closures; however this was not our aim in this study.

The approach suggested in this study for quantifying changes in fishing impact in response to management measures is based on different data sources and involves different assumptions than the more traditional evaluations of fishing impact based on catch information and corresponding population dynamics. Thus, it could both complement the stock assessment based evaluations of management effects and additionally allow addressing changes in fishing impact in situations where reliable stock assessments are not available. New technologies are being introduced in marine fisheries, amongst others to monitor human activities at sea (Kindt-Larsen, 2009; Saitoh et al., 2011). Thus, increasingly detailed new information on fishing activities and related pressures on marine environments is becoming available, which could benefit the analyses of quantifying local fishing impacts based on effort data. Developing methods that use high resolution fisheries data to estimate fishing effects on marine ecosystems is rapidly progressing (e.g., Dinmore et al., 2003; Jennings and Lee, 2012; Lambert et al., 2012). Our study is intended to contribute to this process, demonstrating the use of combined analyses and modeling of fish distribution and fishing pressure in the context of quantifying changes in human impacts and the effects of management measures.

Supplementary material

Supplementary material is available at the ICESJMS online version of this paper. Section A provides additional information on modeling cod distribution and associated uncertainties; section B provides information on estimated fishing effort by gear, area and year.

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Figure captions

Figure 1. Closed areas in the Kattegat. Area 1: seasonally closed area, closed from January 1 to March 31, except for fishery with selective gears; Area 2: partially closed area, closed for all fisheries in the period from January 1 to March 31. Fisheries with selective gears is allowed from April 1 to December 31; Area 3: permanently closed area, closed for all fisheries, including recreational fisheries. Area 4: seasonal closed area in the Northern Sound, closed from February 1st to March 31, except for fishery with selective gears.

Figure 2. Predicted relative distribution of cod by quarter and age. In each panel, the densities are scaled to a mean of 1, by rectangles of 0.01° longitude × 0.01° latitude.

Figure 3. Danish fishing effort from TR2 fleet in the Kattegat by year and quarter. The maps show the sum of VMS hourly pings with vessel speed 2–4 knots within rectangles of 0.05° longitude × 0.05° latitude.

Figure 4. Box and whiskers plots of the distribution of bootstrap replicates of fishing impact on cod in 2009–2011 relative to 2008, by size groups.

Tables

Table 1. The trawl surveys available in the Kattegat by quarter, approximate number of hauls per year and the year range used in the analyses of cod distribution.

Quarter	Survey	Hauls	Years
Q1	International Bottom Trawl Survey (IBTS)	~20	1996–2012
	Baltic International Trawl Survey (BITS)	~22	1996–2012
Q2	No data	–	–
Q3	International Bottom Trawl Survey (IBTS)	~20	2001–2011
Q4	Baltic International Trawl Survey (BITS)	~22	2008–2011
	Danish/Swedish cod survey	~80	2008–2011
	Danish sole survey	min. 70	2008–2011

Table 2. Mean and coefficient of variation (in brackets) of bootstrap replicates of the proportion of cod stock within a given area. Areas 1–3 correspond to closed areas implemented since 2009 (see Figure 1 for definition of areas), Area 0 corresponds to the areas in the Kattegat outside the closures.

Quarter	Size group	Area 0	Area 1	Area 2	Area 3
Quarter 1	10-24 cm	0.75 (0.03)	0.11(0.12)	0.08 (0.12)	0.06 (0.13)
	25-39 cm	0.54 (0.05)	0.20 (0.11)	0.15 (0.12)	0.11(0.15)
	≥40 cm	0.49 (0.08)	0.12 (0.12)	0.19 (0.13)	0.19 (0.12)
Quarter 3	10-24 cm	0.91 (0.02)	0.06 (0.21)	0.03 (0.27)	0.01 (0.35)
	25-39 cm	0.67 (0.09)	0.17 (0.20)	0.14 (0.28)	0.02 (0.40)
	≥40 cm	0.66 (0.16)	0.14 (0.35)	0.18 (0.31)	0.01 (0.71)
Quarter 4	10-24 cm	0.86 (0.01)	0.05 (0.10)	0.05 (0.11)	0.04 (0.11)
	25-39 cm	0.75 (0.02)	0.10 (0.06)	0.10 (0.08)	0.05 (0.10)
	≥40 cm	0.58 (0.04)	0.18 (0.08)	0.20 (0.08)	0.04 (0.13)

Table 3. Fishing impact on cod (by size group) in 2009–2011 relative to 2008, by area. Areas 1–3 correspond to closed areas implemented since 2009 (see Figure 1 for definition of areas), Area 0 corresponds to the areas in the Kattegat outside the closures.

Size	Year	**Area 0**	**Area 1**	**Area 2**	**Area 3**	**Total**
10-24 cm	2008	1.00	1.00	1.00	1.00	1.00
	2009	1.08	1.07	0.11	0.09	0.94
	2010	0.91	1.09	0.30	0.47	0.85
	2011	0.50	0.55	0.09	0.04	0.45
25-39 cm	2008	1.00	1.00	1.00	1.00	1.00
	2009	1.00	1.06	0.06	0.07	0.72
	2010	0.83	1.07	0.11	0.41	0.67
	2011	0.57	0.56	0.04	0.04	0.40
≥40 cm	2008	1.00	1.00	1.00	1.00	1.00
	2009	0.96	1.18	0.18	0.05	0.65
	2010	0.81	1.13	0.30	0.54	0.66
	2011	0.71	0.72	0.11	0.03	0.45

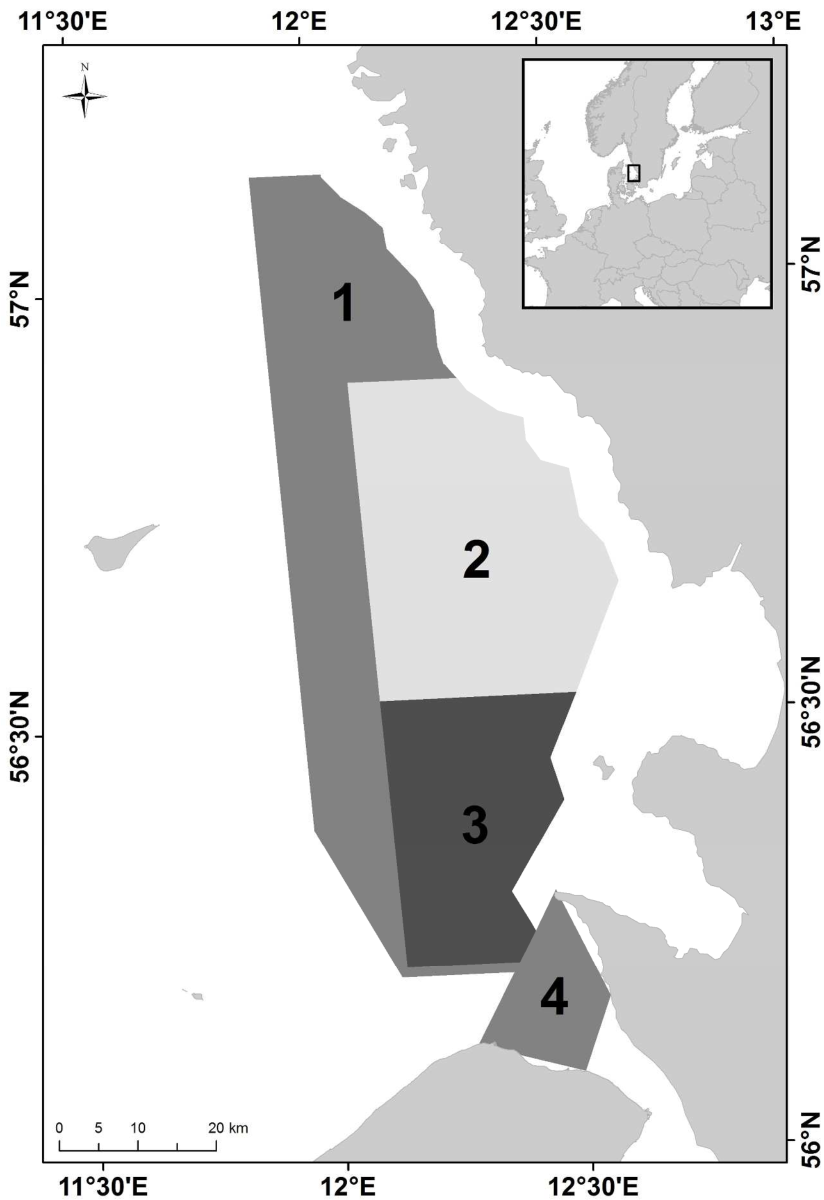

Figure 1
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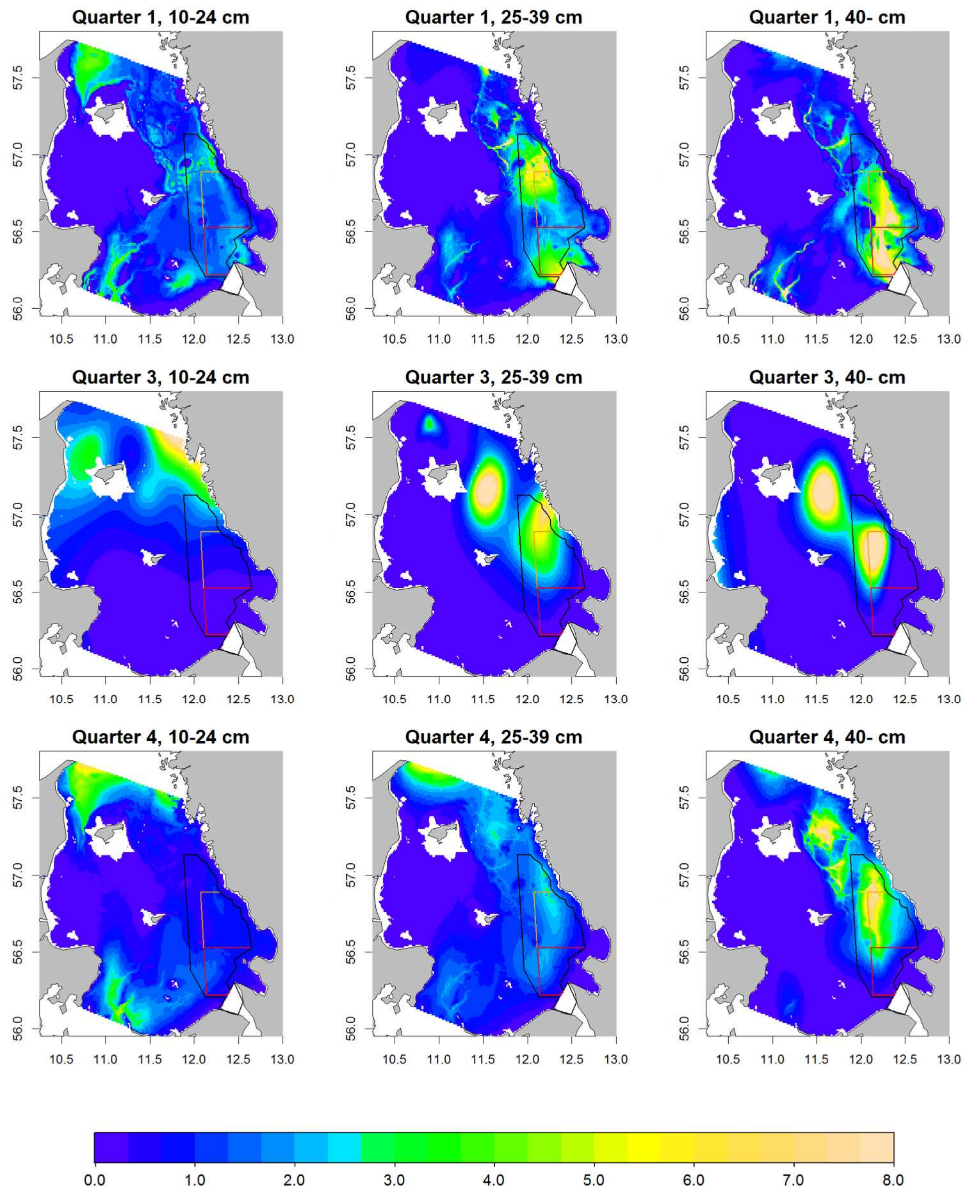

Figure 2
705x846mm (72 x 72 DPI)

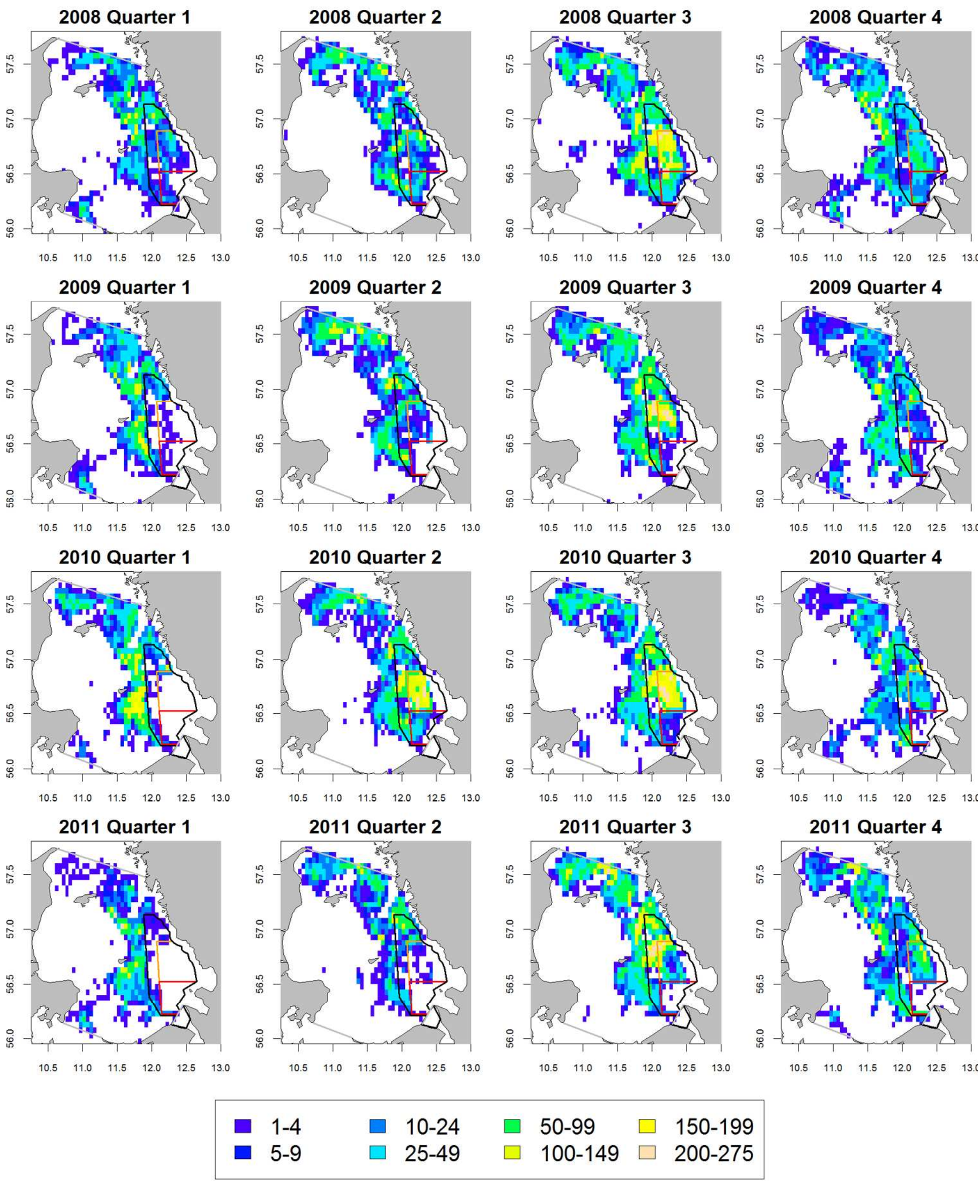

Figure 3
705x846mm (72 x 72 DPI)

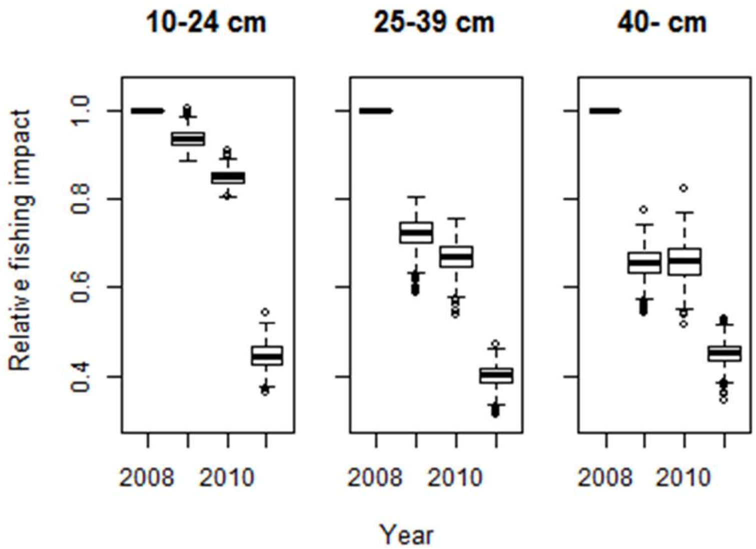

Figure 4
141x105mm (72 x 72 DPI)